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The choice of biological waste treatment method for urban areas in Japan—An environmental perspective

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ABSTRACT

Biological treatment of organic waste is environmentally friendly and a wide range of treatment methods exists. Integrated biological treatment systems with additional equipments, such as pre-treatment, wastewater treatment and deodorisation processes are currently in use. To promote and spread the application of biological waste treatment, a life cycle assessment (LCA) study was conducted on six biological treatment methods: integrated wet anaerobic digestion (AD), integrated dry AD, simple wet AD, simple dry AD, integrated composting and simple composting systems. The impacts of operating rate and wastewater treatment, which affect GHG emissions, were also quantitatively analysed. Integrated wet AD showed the highest total GHG emissions due to the high energy consumption by additional equipments which occupy 80% of the whole process. Integrated composting also presented higher GHG emissions than simple composting because of the higher electricity consumption. Additional equipments are necessary for integrated systems installed in urban areas, and this study suggests that the reduction of energy consumption for these additional equipments is an important issue. Among the additional equipments for AD, wastewater treatment largely affected the GHG emissions. Dry AD normally generates less wastewater due to low moisture content in the waste. Thus, effective treatment of wastes with low environmental loads can be achieved by dry AD, where energy consumption from wastewater treatment is low. On the other hand, methane yield from food waste by dry AD is generally smaller than wet AD. Installing an advanced dry AD reactor with additional functions such as long solid retention time, and adjusting the moisture content of input waste by mixing paper waste will contribute to the efficient treatment of organic waste in urban areas.

Contents

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1. Introduction

Waste treatment is a high energy-consuming process that is in need of improvement in efficiency and reduction in greenhouse gas (GHG) emissions. Organic waste contains large amounts of water and requires a large amount of energy for incineration [1]. Incineration facilities can be deteriorated by heat and acidic gas, and their maintenance costs increase pressure on local governments' finances [2]. Based on these issues, policy reversal from incineration to biological treatment, such as low energy-consuming systems like composting and bio-gasification, is an urgent task.

Biological treatment methods focus on the recyclability of valuable products, such as compost and electricity, in addition to low energy consumption. A number of published papers have studied the environmental impact of waste treatment methods, including landfill, incineration and biological treatments, using life cycle assessment (LCA) methods. A previous study reported GHG emissions from composting processes were 1/30–1/5 that of landfilling of organic waste [3–5]. In comparison studies between composting and incineration, the GHG emission from composting was 1/15–1/2 of incineration [5,6]. In the case of anaerobic digestion (AD) of organic waste, reports show the GHG emissions are approximately 1/3 of landfill [7] and approximately 3/5 of incineration [4]. Based on these studies, biological waste treatment methods are considered more environmentally friendly than incineration or landfilling. The low energy consumption and low environmental impact are attributed to the aerobic and anaerobic decomposition of organic matter by bacteria in the main process of waste treatment.

Among the biological treatment methods available today, AD is gaining attention in the effective treatment of low quality food waste and its competent biogas yield which can be used directly for electric power generation. Despite the benefits, food waste treatment by AD is far from a full-fledged operation in Japan. The Japanese government has encouraged the use of biomass through the Fundamental Law for Establishing a Sound Material-Cycle Society enacted in 2001, the Food Recycling Law enacted in 2001 (revised in 2007), the New Energy Law enacted in 1997 (revised in 2008) and the Law for the Promotion of Utilisation of Biomass enacted in 2009. Takata et al. [8] studied the effect of the Food Recycling Law and evaluated its advantages and drawbacks. Various food waste recycling methods have become widespread after the Food Recycling Law enforcement, and currently the recycling rate of food waste in the manufacturing sector is over 90% while the recycling rate of the food service industry is under 20% [9] because of the characteristically low quality waste containing high salt and oil content.

On the other hand, AD treatment has been implemented and successfully expanded as part of Europe's energy policy [10,11]. In Germany, where the use of organic waste for energy renewal is promoted, there were 100 anaerobic digestion (AD) facilities in 1990.

This number increased to 3400 in 2006 due to the Renewable Energy Promotion Law enacted in 2000 which had energy production as one of the goals [12]. Furthermore, the usage of waste is popularised by the Feed-in Tariff which oblige power companies to buy electricity produced by renewal energy in Germany [13]. In Denmark, higher treatment fees for incineration and landfills, personnel training relevant to waste management and long-term financial support have led to a successful increase in the AD treatment of organic waste [10,14].

Thus, AD is known as a suitable and practical method to treat organic waste. However, it has a complex process and does not exist independently, and requires additional equipments, such as pre-treatment, wastewater treatment, and composting process. There is a wide variety of AD systems such as UASB which treats sewage water with low total solid (TS), wet system which treats waste with 5% of TS [15] and dry system with TS at 25–40% [16]. The required additional equipments are different among these systems as well as the aggregate environmental impact and operation cost. In this paper, we focused on the impact of additional equipments on various AD processes in the recycling of low quality food waste.

In addition, upgraded additional equipment is necessary to manage the complex waste composition and to prevent the release of offensive odour and wastewater in densely populated areas, For successful biological waste treatment and effective usage of biomass in urban areas, energy consumption and GHG emissions, including the additional equipment needed in waste treatment facilities, should be investigated.

The objective of this study is to evaluate the GHG emissions of various AD and composting systems and to identify the impact of additional equipment on the processes. To clarify the environmental impact of additional equipments, which is essential for urban waste management, the integrated and simple systems of wet and dry type of biological treatments were examined. This study also considered both actual and predicted GHG emissions in the planning phase of a waste treatment facility.

2. Methods

An evaluation of the environmental impact of six biological treatment systems – integrated wet AD, integrated dry AD, simple wet AD, simple dry AD, machine-integrated composting and conventional composting – was conducted by LCA. Data on energy usage for waste treatment and the amount of recycled products were collected from interview surveys in 2007 (Table 1). CO_2, CH_4 and N_2O production were considered, and biogenic CO_2 was not included in GHG because it was considered carbon neutral in this study. The GHG emission factors are listed in Table 2. The equivalent global warming potentials of CH_4 and N_2O

Table 1
Material and energy balance.

Scenario	Parameter	Value	Unit	Source	Output	Value	Unit	Source
S1	Energy input:				Biogas yield	166	N m^3/t-waste	Interview
	Pre-treatment	139	kWh/t-waste	Interview	Power generation at 27% of generating efficiency	230	kWh/t-waste	
	Anaerobic digestion	20	kWh/t-waste		Power generation at 33% of generating efficiency	283	kWh/t-waste	
	Energy generation	6	kWh/t-waste		Compost	100	kg/t-waste	
	Wastewater treatment	112	kWh/t-waste		CH_4 in biogas	52	%	
	Composting	51	kWh/t-waste		N in compost	4.16	%	[17]
	Deodorisation	43	kWh/t-waste		P in compost	5.3	%	
	Dilution water	0.9	t/t-waste		Water content of compost	33.5	%	
S2	Energy input:				Biogas yield (incl. paper)	205	N m^3/t-waste	[19]
	Anaerobic digestion	75	kWh/t-waste	[18]	Biogas yield (food waste)	150	N m^3/t-waste	
	Wastewater treatment	39	kWh/t-waste		Power generation (incl. paper)	413	kWh/t-waste	
					Power generation (food waste)	284	kWh/t-waste	
					CH_4 in biogas	57.8	%	
					Solid residue	83	kg/t-waste	[20]
					Water content of solid residue	33	%	[18]
S3	Energy input:				Biogas yield	166	N m^3/t-waste	Interview
	Anaerobid digestion	20	kWh/t-waste	Interview	Power generation	230	kWh/t-waste	
	Energy generation	6	kWh/t-waste		CH_4 in biogas	52	%	
	Dilution water	0.9	t/t-waste		Liquid fertiliser	2.35	t/t-waste	
S4	Energy input:				Biogas yield (incl. paper)	205	N m^3/t-waste	[19]
	Anaerobic digestion	75	kWh/t-waste	[18]	Biogas yield (food waste)	150	N m^3/t-waste	
	Composting	0.2	kWh/t-waste	Interview	Power generation (incl. paper)	413	kWh/t-waste	
		6.6	l/t-waste		Power generation (food waste)	284	kWh/t-waste	
					CH_4 in biogas	57.8	%	
					Compost production	83	kg/t-waste	[20]
					Water content of compost	33	%	[18]
S5	Energy input:	115	kWh/t-waste	Interview	Compost production	125	kg/t-waste	Interview
	Pruning waste	125	kg/t-waste		N in compost	3.9	%	
	Ethanol	0.8	kg/t-waste		P in compost	1.25	%	
					Water content of compost	13.4	%	
S6	Energy input:				Compost production	318	kg/t-waste	Interview
	Electricity	0.2	kWh/t-waste	Interview	N in compost	1.5	%	
	Diesel	6.6	l/t-waste		P in compost	0.5	%	
	Pruning waste	333	kg/t-waste		Water content of compost	40	%	

were 23 kg-CO_2/kg-CH_4 and 296 kg-CO_2/kg-N_2O over a 100-year time scale [28]. Recycled products, such as biomass-derived electricity and compost, were considered substitutes for electricity and chemical fertilisers. Hence, GHG savings by material recycling were subtracted from the GHG emissions of each system.

2.1. Functional unit and system boundaries

The waste composition of each biological treatment system is shown in Fig. 1. The GHG emissions from each treatment of one ton of organic waste (including paper in dry AD) and one ton of food waste were calculated.

The six systems in this study are presented in Fig. 2. Waste treatment processes from pre-treatment to the production of recycled products or final disposal were included. The environmental assessment of the construction phase was limited because biological treatment facilities have a long service life, and GHG emissions at the construction phase are negligible [29].

2.2. Scenario design

2.2.1. Scenario 1 (S1): Integrated wet AD

This system is operated by a wet type (more than 90% of moisture content) with mesophilic digestion (35–37 °C). The main waste in S1 is food waste, which comprises approximately 80% of total waste. Paper is not inputted. The operating process involves pre-treatment (waste separation and crushing), the main process (AD), wastewater treatment, composting, energy recovery from biogas, and deodorisation (Fig. 2(S1)).

After the main process, the solid and liquid digestive fluids are separated. The solid is recycled to compost, and the liquid is denitrified to meet the Japanese effluent standard (160 mg-COD/l) and then released to sewage. The waste is radically screened of harmful substances, and its residue is recycled to quality compost. An actual power-generating efficiency of 27% was used to calculate actual GHG emissions. To compare the GHG emissions of one ton of food waste with the other scenarios, a generating efficiency of 33%, as in S2, was adopted.

The operating rate of this facility is 43% (24 t/day) against a capacity of 55 t/day. Therefore, GHG emissions were calculated at both 43% and 100% operating rate based on the estimate equation presented in Table 3.

2.2.2. Scenario 2 (S2): Integrated dry AD

This system is operated by a dry type (60–85% of moisture content) with thermophilic digestion (55 °C) [19,20]. The treated waste includes approximately 70% food waste and approximately 30% paper waste. Biogas from paper waste was excluded from the calculation of GHG emissions for one ton of food waste. The operating process involves simple pre-treatment (metal removal only), the main process (AD), wastewater treatment, composting and energy recovery by bio-gasification (Fig. 2(S2)). Although no wastewater is released from dry AD normally, fermentation residue is dewatered and the treated water is used in this facility. Electricity is generated by a power generator with a generating

Table 2
Inventory data.

Process	GHG	Value	Unit	Source
Power production:	CO_2	0.425	kg-CO_2/kW h	[21]
	CH_4	0.00	g-CH_4/kW h	
	N_2O	0.0021	g-N_2O/kW h	
Diesel production:	CO_2	0.102	kg-CO_2/l	[22]
	CH_4	0.0744	g-CH_4/l	
	N_2O	0.0444	g-N_2O/l	
Diesel combustion:	CO_2	2.73	kg-CO_2/l	
	CH_4	0.0744	g-CH_4/l	
	N_2O	0.0444	g-N_2O/l	
Kerosene production:	CO_2	0.123	kg-CO_2/kg	
	CH_4	0.0897	g-CH_4/kg	
	N_2O	0.0535	g-N_2O/kg	
Kerosene combustion:	CO_2	2.59	kg-CO_2/l	
	CH_4	0.0708	g-CH_4/l	
	N_2O	0.0423	g-N_2O/l	
Density:Kerosene	–	0.8	kg/l	
Heavy oil A production	CO_2	0.107	kg-CO_2/l	[22]
	CH_4	0.0780	g-CH_4/l	
	N_2O	0.0466	g-N_2O/l	
Heavy oil A combustion	CO_2	2.82	kg-CO_2/l	
	CH_4	0.0780	g-CH_4/l	
	N_2O	0.0466	g-N_2O/l	
Chemical fertiliser production	CO_2	3.56	kg-CO_2/kg-N	[23]
	CH_4	0.00	g-CH_4/kg-N	
	N_2O	0.02	g-N_2O/kg-N	
Chemical fertiliser production	CO_2	0.10	kg-CO_2/kg-P	[24]
	CH_4	0.00	g-CH_4/kg-P	
	N_2O	0.00	g-N_2O/kg-P	
Reduction from nitrogenous fertiliser	N_2O	3.158	kg-CO_2/kg-N	[24]
Heat conversion rate of electricity	–	860	kcal/kW h	[25]
Residue incineration:				[26]
Electricity	–	0.10753	kW h/kg	
Kerosene	–	0.000221	l/kg	
Heavy oil A	–	0.002708	l/kg	
CH_4 emission	CH_4	0.00	g-CH_4/kg	
N_2O emission	N_2O	0.088	g-N_2O/kg	
Lower heating value	–	1300	kcal/kg	
Efficiency of power generation	–	10	%	
Wastewater treatment process	N_2O	0.026	kg-N_2O/t	[27]
Composting	CH_4	0.052	kg-CH_4/t	
	N_2O	0.061	kg-N_2O/t	

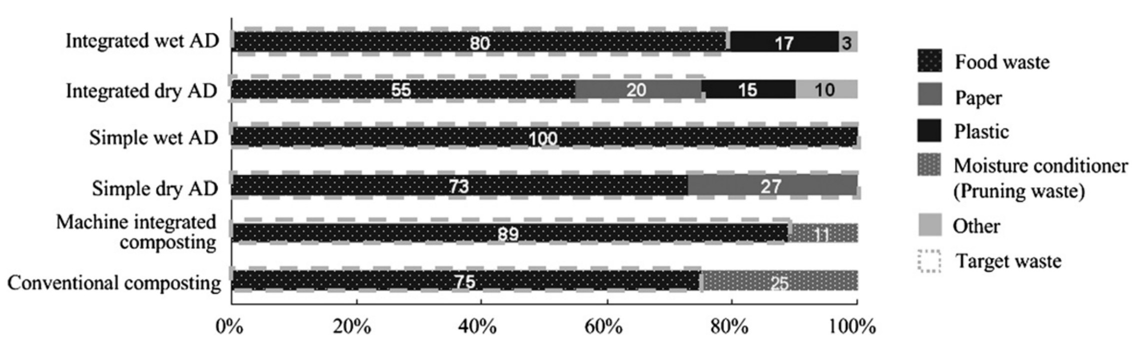

Fig. 1. Waste composition of each facility.

efficiency of 33%. The energy consumption of the pre-treatment, deodorisation, composting and energy recovery processes was counted as part of the treatment.

2.2.3. Scenario 3 (S3): Simple wet AD

This system is operated by a wet type (more than 90% of moisture content) with mesophilic digestion (35–37 °C). In this system, waste is anaerobically digested without a pre-treatment process (Fig. 2(S3)). In S3, only food waste is treated, as in S1. The digestive fluid is used as liquid fertiliser, and wastewater treatment is not required. Deodorisation equipment is also not necessary because there are no residences near the facility. This type of facility is used only in a region of northern Japan.

The actual generating efficiency of the power generator was 27%, which was used to calculate the actual GHG emissions. A generating efficiency of 33% was used to compare the GHG emissions of one ton of food waste to the other scenarios.

2.2.4. Scenario 4 (S4): Simple dry AD

Waste is anaerobically digested without a pre-treatment process (Fig. 2(S4)). The fermenter's temperature and the water

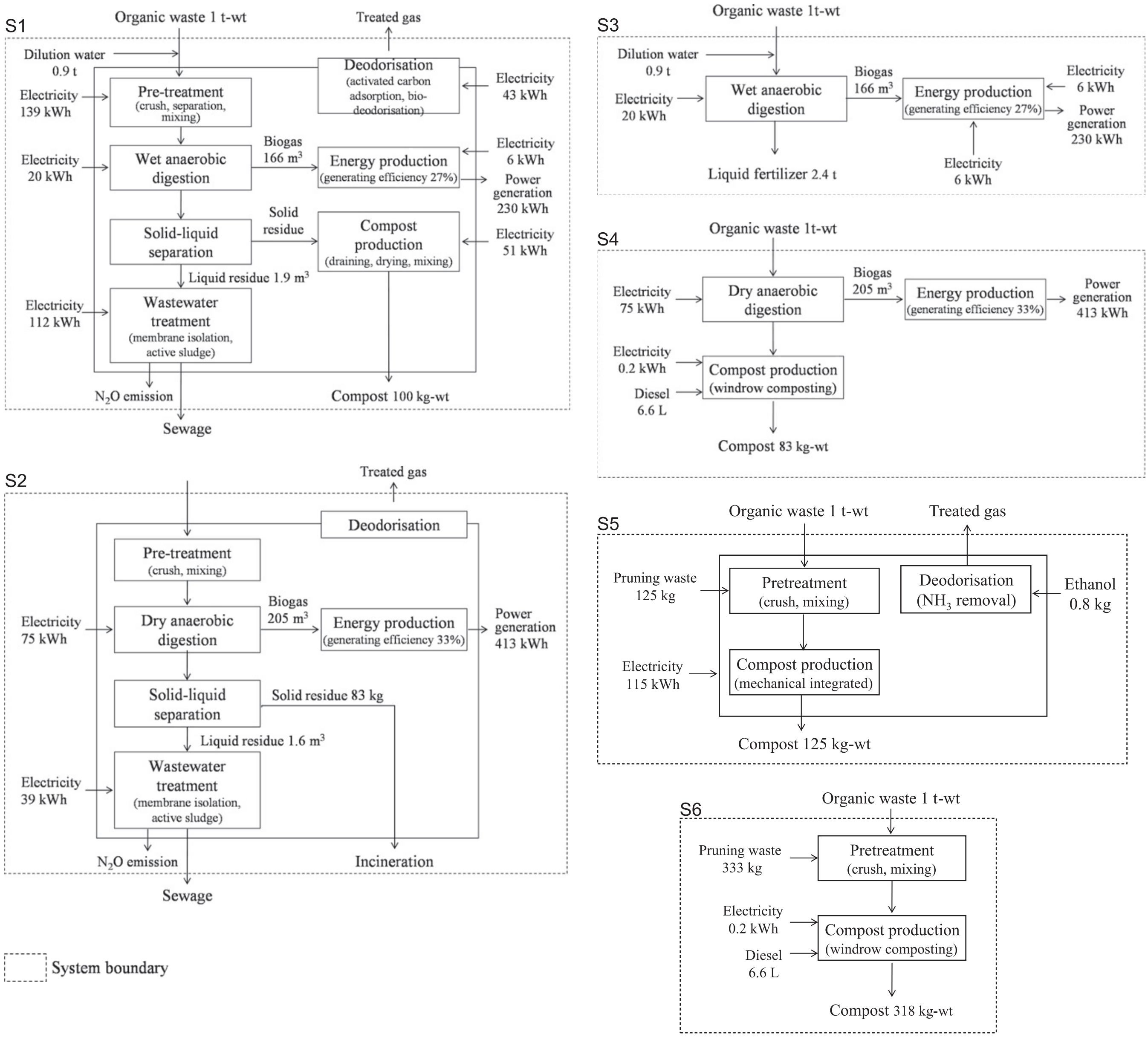

Fig. 2. Scenario description. (S1) Integrated wet AD, (S2) Integrated dry AD, (S3) Simple wet AD, (S4) Simple dry AD, (S5) Machine integrated composting and (S6) Conventional composting.

content of the waste are the same as S2. The treated waste includes approximately 70% food waste and 30% paper waste. Biogas from paper waste was excluded from the calculation of GHG emissions for one ton of food waste. Deodorisation equipment is not needed because there are no residences near the facility. Digestive fluid is used as a moisture conditioner for compost, and no wastewater treatment is needed. Compost is produced with periodic aeration.

2.2.5. Scenario 5 (S5): Machine-integrated composting

In this system, the entire composting process is managed by computer. The treatment process involves pre-treatment, composting and deodorisation (Fig. 2(S5)). Approximately 90% food waste and 10% pruning waste is treated in S5. Food waste is crushed and mixed with moisture conditioner (pruning waste) in the pre-treatment process. The mixed matter is then poured into a lattice-shaped container that is $1555 \times 1555 \times 1020$ cm in size. The mixed matter is aerated by moving it from one container to another. This facility owns 2700 containers. The temperature and moisture condition of the compost are monitored by computer.

The energy consumption of the pre-treatment and composting processes was counted as part of treatment. The energy consumption of deodoriser was calculated as 15 kW with 24 h of operation.

2.2.6. Scenario 6 (S6): Conventional composting

This is the most common composting system in Japan. S6 treats 75% food waste and 25% pruning waste. In this system, food waste and pruning waste are periodically mixed by heavy machines (Fig. 2(S6)). The compost is then piled and matured.

Table 3
Estimate equitation of S1 for operating rate 100% (Capacity: 55 t/day, actual amount of treatment: 24/day).

Process	Measured value (operating rate 43%) (a)	Predicted value (operating rate 100%) (b)	Equipments	Method for estimating
Pre-treatment	139 kW h/t	61 kW h/t	– Crusher (Pulper) – Separator (Multi sorter) – Solubilisator	It is operated regardless of the amount of waste (8 h/day) and the energy efficiency increases when the the facility is fully operated.($b=a\times24/55$)
Anaerobic digestion	20 kW h/t	9 kW h/t	– BIMA, no engine mixer	It is operated regardless of the amount of waste and the energy efficiency increases when the the facility is fully operated. ($b=a\times24/55$)
Wastewater treatment	112 kW h/t	49 kW h/t	– Membrane separator – Acrivated carbonadsorption – Biological denitrification	It is operated regardless of the amount of waste and the energy efficiency increases when the the facility is fully operated. ($b=a\times24/55$)
Deodorisation	43 kW h/t	19 kW h/t	– Biological deodoriser – Activated carbonadsorption	It is operated regardless of the amount of waste and the energy efficiency increases when the the facility is fully operated. ($b=a\times24/55$)
Composting	51 kW h/t	51 kW h/t	– Dehydrater – Dryer – Aerator (shovel loaders)	Operating time of shovel loaders is proportionate to the amount of waste. ($b=a$)
Electricity generator	6 kW h/t	6 kW h/t	– Desulfurisator – Dual fuel gas generator	One generator is operated if the amount of waste is small while three generators are operated when the the facility is fully operated. ($b=a$)
Biogas yield Compost production	166 N m^3/t 100 kg/t	166 N m^3/t 100 kg/t		Biogas yield is proportionate to the amount of waste. ($b=a$) Compost production is proportionate to the amount of waste. ($b=a$)

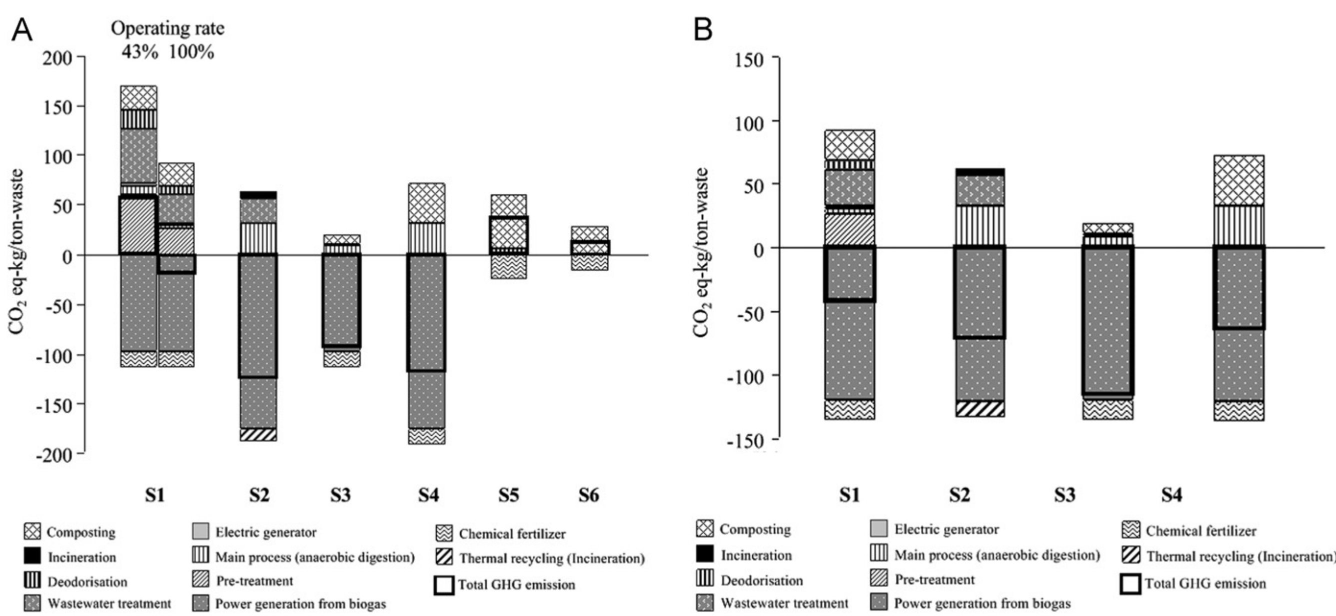

Fig. 3. (A). GHG emissions for each scenario. ((S2) and (S4) include paper waste). (B). GHG emissions from the treatment of food waste.

2.3. Comparison of actual and predicted value.

The actual and predicted GHG emissions/reductions in the planning phase of S1 and S2 were surveyed. The operating rate, energy consumption, amount of energy recovery and end-usage of fermentation residue were compared.

2.4. The amount of biogas yield by AD

In LCA study, GHG reduction of AD is largely affected by biogas yield, and the data of yield amount are obtained from interview, literature, or calculation by theoretical calorific value and chemical composition. It was reported that the actual amount of biogas

is less than estimated by calculation [30]. Therefore, further examination of biogas yield was made through the comparison between LCA and experimental studies.

3. Results

3.1. Outline of LCA results

The total GHG emissions per one ton of organic waste (including paper) are shown in Fig. 3(A). The operating rate of S1 was low (43%), whereas the other facilities' operation rates were approximately 100%. To compare all scenarios under identical conditions, the GHG emissions for 100% of the operation rate were estimated in S1 (the right bar). The black frame represents the total GHG emissions.

The lowest GHG emissions occurred in S3 (simple wet AD), followed by S2 (integrated dry AD), S4 (simple dry AD), S5 (conventional composting), S1 (integrated wet AD) and S5 (machine-integrated composting).

Three AD scenarios (S2, S3 and S4) showed lower GHG emissions than composting because of large GHG reductions due to energy production from biogas. The highest total GHG emissions of S1 (54 kg-CO_2eq/t-waste) was a result of the additional equipment and low operating rate. The total GHG emissions of S1 dropped to −21 kg-CO_2eq/t-waste when the operating rate was 100%.

GHG emissions from the composting process of AD systems were relatively high in S1, S3 and S4, and the wastewater treatment process had a large environmental impact in S1 and S2.

In the composting scenarios, the GHG emissions of S5 were high (61 kg-CO_2eq/t-waste) because of electricity consumption. The GHG emissions of S6 had a lower environmental impact of 29 kg-CO_2eq/t-waste.

3.2. The environmental impact of additional equipment

In the comparison of the integrated and simple systems, the GHG emissions of S1 were higher than S3, and the difference was notably large (74 kg-CO_2eq/t-waste) (Fig. 3(A)). The GHG emissions from additional equipment in S1, including pre-treatment, wastewater treatment and deodorisation, were 70% of the entire GHG emissions in S1. High GHG emissions from wastewater treatment were also presented in S2.

S4 showed the second-highest GHG emissions (72 kg-CO_2eq/t-waste) derived from the composting process. However, S4 requires no wastewater treatment system due to the low water content of the waste and therefore it needs no energy for wastewater treatment.

A low demand for fertiliser in urban areas and Japan's strict effluent standard leads to high energy consumption for wastewater treatment in the integrated systems. The energy reduction from additional equipment is the key to the promotion of AD in urban areas.

3.3. The environmental impact between wet and dry AD

The dry systems (S2 and S4) showed remarkably lower total GHG emissions than the wet systems (S1 and S3) due to the large amount of energy recovery from paper waste (Fig. 3(A)). The simple dry AD (S4) also showed lower total GHG emissions than the simple wet AD (S3). Dry AD of paper waste is an environmentally preferable system because gas yield of paper is higher (490 N m^3/t) than food waste (110–160 N m^3/t) [31], and since urban waste contains significant amounts of paper waste [32] dry AD can be an efficient waste treatment method for urban wastes.

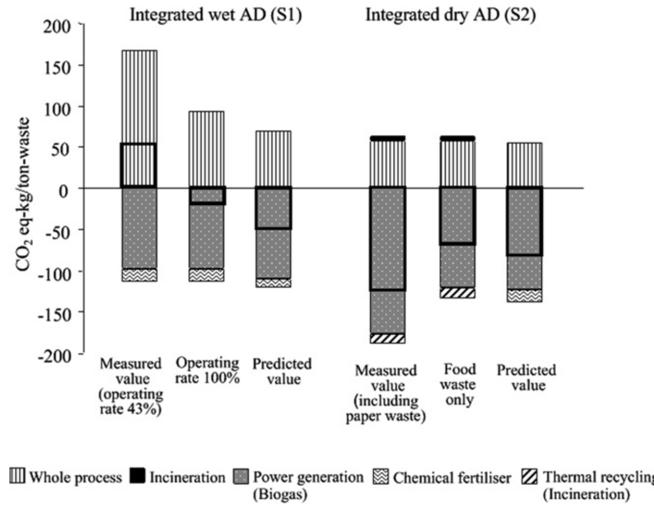

Fig. 4. GHG emissions by actual operating and predicted values.

The total GHG emissions of food waste treatment and the GHG reduction of all treatment scenarios were similar at approximately −120 kg-CO_2eq/t-waste (Fig. 3(B)). S3 showed the lowest total GHG emissions (−116 kg-CO_2eq/t-waste) because of the low energy consumption of the treatment. However, the actual fertilisation efficiency of liquid fertiliser produced by wet AD is lower than recycled compost [33], which may lead to lower GHG reduction, and this needs to be taken into consideration in the future.

From these results, the dry system is found to be effective when treating organic waste containing paper but GHG reduction is comparable with wet AD when treating only food waste.

3.4. The impact of the difference between actual and predicted values

The actual and predicted values of the integrated AD (S1 and S2) were compared. The actual value of S1 at 43% operation rate showed notably higher total GHG emissions (54 kg-CO_2eq/t-waste) than the predicted value (−53 kg-CO_2eq/t-waste) (Fig. 4). At 100% operation rate, the total GHG emissions was predicted to be −21 kg-CO_2eq/t-waste. The comparison of actual and predicted values showed that GHG emissions becomes higher when the amount of waste becomes lower than the capacity of AD reactor.

The difference between the actual and predicted values of S1 is due to the operating time. The pre-treatment, main process and wastewater treatment were carried out regardless of the amount of waste. Thus, energy consumption was larger when the amount of waste is small.

In S2, the actual total GHG emissions (including paper) were lower (−125 kg-CO_2eq/t-waste) than the predicted value (−84 kg-CO_2eq/t-waste) because paper waste was not included in the plan. On the other hand, the fermentation residue was supposed to be recycled to compost in the planning phase, but the residue was actually incinerated, thus GHG reduction was not obtained from composting.

4. Discussion

4.1. Methane yield (m^3-CH_4/t-VS) comparison between LCA and experimental studies

In AD process, factors such as energy consumption from pre-treatment, main process or wastewater treatment, and energy recovery from biogas directly affect the total GHG emissions.

Table 4
Comparison of methane yield (m^3CH_4/t-VS) between LCA/EA and experimental studies.

Reactor type	Waste type	Study type	TS (%)	VS (%)	Methane yield (m^3-CH_4/t-VS)	Reference
Wet	Food	LCA	29	91	326	This study
Wet	Food	LCA	20	85	302	[36]
Wet	Food	LCA	44	n.d.	380	[8]
Wet	Food, other	LCA	33	n.d.	450	[34]
Wet	Food, yard	LCA	33	85	313	[37]
Wet	Food, paper, yard, other	LCA	45	80	130	[38]
Wet	MSW	LCA	30	n.d.	428	[39]
Wet	Farmyard, cow manure	LCA	8	85	146–160	[40]
Wet	Pig manure	LCA	4	76	360	[41]
Wet	Sewage sludge	EA	n.d.	84	218	[42]
Wet	Food	EXP	8–31	63–95	266–482	[30,43–47]
Wet	MSW	EXP	13–48	70–95	222–450	[30,45,48,49]
Wet	Yard	EXP	5–15	25–83	209–452	[30,45,50]
Wet	Crop	EXP	19–49	72–98	16–412	[30,51–56]
Wet	Pig manure	EXP	4–29	64–85	150–492	[54,57–63]
Wet	Cow manure	EXP	6–12	69–90	74–250	[52,61,64,65]
Wet	Sewage sludge	EXP	n.d.	n.d.	274	[66]
Dry	Food, paper	LCA	55	92	233	This study
Dry	Food, paper	LCA	44	86	152–192	[32]
Dry	Food	LCA	19	85	689	[67]
Dry	Food	LCA	23–43	n.d.	575	[68]
Dry	Food, yard	LCA	35	n.d.	540	[69]
Dry	Food, yard	LCA	33	86	301	[35]
Dry	Food, paper, yard, other	LCA	45	89	68	[38]
Dry	Pig manure, crop, glycerol	EA	48	88	503	[70]
Dry	Food	EXP	10–38	69–89	200, 216	[30,71]
Dry	Food, paper	EXP	n.d.	95	270	[72]
Dry	MSW	EXP	17–30	43–95	60–290	[48,73–77]
Dry	Yard	EXP	92–98	89–95	41–165	[78–80]
Dry	Crop	EXP	10–98	77–97	67–280	[30,52,78,81,82]
Dry	Dairy waste	EXP	7–11	63–82	107, 239	[71,81]
Dry	Sewage sludge	EXP	10	73	115	[71]

LCA: Life cycle assessment, EA: Energy assessment, EXP: Experimental data.

Among these factors, biogas yield have a large impact on GHG reduction. Previous studies have shown that GHG reduction by biogas-derived energy accounted for most of the total reduction at 66% [34], 85% [35] or 92% [4]; data of [4] and [34] were estimated from figures in their literature. The amount of biogas yield from AD is critical for the estimation of GHG emission. However, it is less known that the biogas yield is different between wet and dry AD, and the amount of biogas production from dry AD is about half of wet AD (Table 4). In dry AD, the total solids (TS) concentration of digested matter is high, while intermediate products and volatile fatty acids (VFA) cannot be sufficiently sent to archaea which create methane gas. Consequently, the organic waste is not digested and methane production decreases [15].

Despite low methane production through the dry AD process, previous LCA studies used high values such as 689 m^3-CH_4/t-VS [67], 575 m^3-CH_4/t-VS [68] and 540 m^3-CH_4/t-VS [69] based on the volatile solids (VS). There are two main reasons for these high values. First, municipal solid waste (MSW) is a mixture of organic waste, paper and yard waste, and when treated, the proportion of methane production from paper and yard waste is considered. Second, methane production is estimated from chemical composition such as carbon, nitrogen or hydrogen, and the value is obtained when the organic matter is totally digested. In the wet AD process, nearly 90% of organic matter is digested and approximately 450 m^3-CH_4/t-VS can be obtained. In contrast, the degradation rate of paper and yard waste is generally low, and only 50% of organic waste is digested. Although it is recently reported that the degradation rate increases by co-digestion, the rate does not improve significantly [83,84]. It is also reported that the methane production is determined by the degradation rate of each organic matter [30].

Methane production from food waste treatment is up to 500 m^3-CH_4/t-VS in wet AD, and it would be appropriate to use 450 m^3-CH_4/t-VS in LCA studies. In the case of dry AD, methane production does not exceed 300 m^3-CH_4/t-VS by treating food waste, and using 250 m^3-CH_4/t-VS will be suitable for the assessment of GHG reduction from biogas yield.

4.2. Influential factors of the total GHG emissions from biological treatment

In LCA study of waste recycling, the GHG emissions are calculated from energy consumption during the recycling process, and the GHG reduction is estimated from the amount of recovered products/energy. An accurate estimation of both GHG emissions and reduction is important because the total GHG emissions is evaluated by subtracting these values.

As stated in the previous section, biogas yield has a great impact on the GHG emissions. Nevertheless, (1) pretreatment such as segregation and crushing of waste, (2) necessity of wastewater treatment, and (3) operating rate affect the total GHG emissions of AD process. To discuss the factors that have an impact on the GHG emissions/reduction in the actual process, the results of this study and the literature data are summarized in Fig. 5. The horizontal axis represents the GHG emissions, and vertical axis shows the GHG reduction from electricity generation or composting. The LCA results located above the diagonal line indicate environmental inefficiency while the results below the line indicate environmental efficiency. The total GHG emissions are ± 0 when $Y=X$.

All composting systems are located above the diagonal line and showed similar efficiencies. In composting systems, there are fewer factors affecting the GHG emissions compared to AD. For

example, frequency of aeration and mixing of compost increases/decreases in proportion to the amount of waste, and the energy consumption accordingly changes. In addition, composting is a simple process which needs no energy-consuming additional facilities such as wastewater treatment. Therefore, GHG emissions do not vary among each composting system. Machine-integrated composting systems showed slightly higher GHG emissions than conventional systems perhaps due to the energy consumption by the automated facility.

On the contrary, the GHG emissions/reduction varied widely among AD systems due to the following factors: (1) operating rate, (2) wastewater treatment and/or (3) methane yield from one ton-wet of organic waste in the AD system. In AD systems, an anaerobic condition has to be maintained in the reactor and energy is required for heat retention and mixing regardless of the waste amount. High total GHG emissions occur if operating rate is low because biogas yield is small. The GHG emissions calculated from the energy consumption of three AD facilities with low operating rate (46–61% reported by Tanikawa et al. [86] indicated by "*f*" in Fig. 5, were higher than the AD facilities at 100% operating rate. The wet AD of this study with operating rate of 40% showed a similar tendency of high GHG emissions.

Next, wastewater treatment was found as a critical factor affecting GHG emissions. The fermentation residue contains high chemical oxidation demand (COD), and therefore wastewater treatment is required before the residue is released. A large amount of wastewater is generated especially in wet AD, and the energy consumption for its treatment leads to high GHG emissions. Previous studies of Börjesson and Berglund [87] and Clavreul et al. [34], indicated in Fig. 5 as "*i*" and "*j*", respectively, showed remarkably low GHG emissions without taking into account wastewater treatment, while AD with wastewater treatment showed higher GHG emissions. Although S1 with operating rate of 100% showed lower GHG emissions than composting and wet AD with low operating rates, the GHG emissions was higher than that of wet AD without wastewater treatment. In urban areas, wastewater treatment is essential because of the low demand for fertilizers produced from the fermentation residue after the wet AD process. Therefore, energy consumption of wastewater treatment should be considered when wet AD is installed in urban areas.

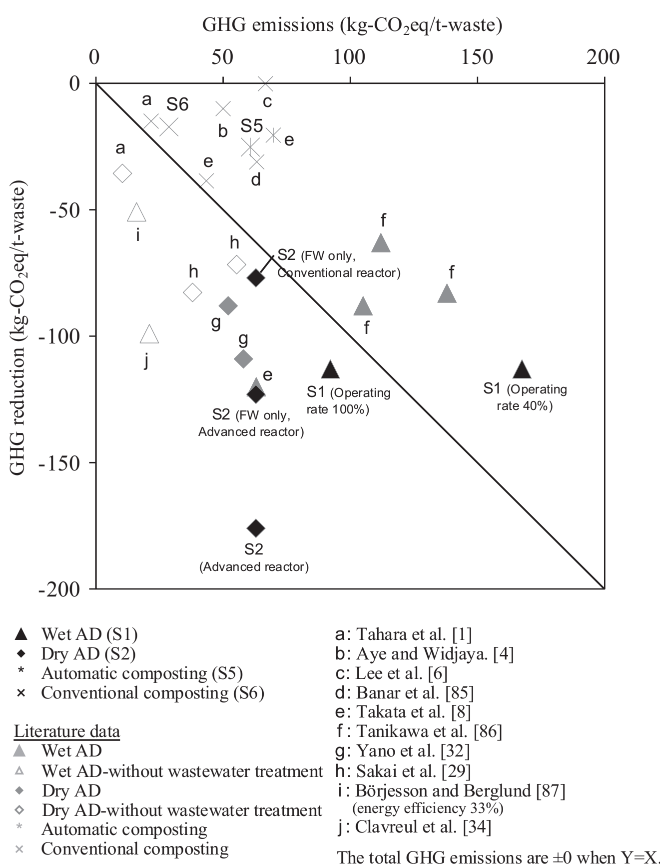

Fig. 5. Comparison with literature data.

Finally, the impact of methane yield on GHG emissions is discussed as a considerable factor. The GHG reduction in LCA study is calculated by the methane yield per waste weight (m^3-CH_4/t-waste). This value is different from the methane yield discussed in 4.1. (m^3-CH_4/t-VS), and the weight is measured as wet weight. Furthermore, the methane yield was different from facility to facility depend on the waste type, and the LCA results varied widely. Tahara et al. [1] calculated the value of food waste treatment by dry AD and found it collects less methane and showed a low GHG reduction. In AD, less GHG reduction is obtained from food waste treatment because of high moisture and low VS content in one ton-wet of organic waste. Despite the low methane yield of dry AD, Yano et al. [32], indicated as "*g*" in Fig. 5, showed a relatively high GHG reduction. This indicates that the low moisture content and high VS in one ton of organic waste leads to an increase in biogas production. Low moisture content of food waste comprising a large amount of bread, achieved high GHG reduction in "*e*" of Takata et al. [8].

A specific operating procedure for dry AD to obtain high methane yield is also recently reported. In S2, the methane yield is high due to the long solid retention time (SRT) attained by returning the TS back into the reactor after extracting its water in the digestion chamber. It is indicated that biogas is efficiently collected by installing such an advanced reactor. The efficient treatment of urban waste, which includes a large amount of paper waste [32], will be possible by dry AD.

5. Conclusion

This study conducted a comparison of integrated and simple biological waste treatment methods, wet/dry AD systems and composting systems to investigate the impact of additional equipments through a LCA method. The impacts of operation rate and wastewater treatment which affect the GHG emissions were also analysed.

The results showed that the total GHG emissions of AD were generally lower than composting. However, total GHG emissions were higher than composting when the AD operating rate is low.

High total GHG emissions due to additional equipments such as pre-treatment, deodorisation and wastewater treatment facilities were also found in the integrated wet AD (62 kg-CO_2eq/t-waste, 100% operating rate). The simple wet AD system, which needs no additional equipments, had the lowest GHG emissions (20 kg-CO_2eq/t-waste). The integrated composting system showed the second-highest total GHG emissions (35 kg-CO_2eq/t-waste) due to electricity usage while the total GHG emissions of the simple system were lower at 12 kg-CO_2eq/t-waste.

Among the additional equipments of AD, wastewater treatment largely affected the GHG emissions. Dry AD normally produces less amount of wastewater due to low moisture content in the waste. Waste treatment with low environmental load can be achieved effectively by dry AD which has a low energy consumption from wastewater treatment.

On the other hand, methane yield from food waste treatment by dry AD is generally smaller than wet AD. Installing an advanced dry AD reactor and mixing paper waste to adjust

moisture content in the input waste, will help the efficient treatment of organic waste in urban areas.

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